

# Demographic trends in Claremont California's street tree population

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## ABSTRACT

The aim of this study was to quantify street tree population dynamics in the city of Claremont, CA. A repeated measures survey (2000 and 2014) based on a stratified random sampling approach across size classes and for the most abundant 21 species was analyzed to calculate removal, growth, and replacement planting rates. Demographic rates were estimated using a hierarchical Bayesian framework. The community-level (all species) median growth rate was 1.41% per year (95% CI: 1.21–1.65%) with *Pinus brutia* and *Pistacia chinensis* growing significantly faster than the community-level median. The community-level median removal rate was 1.03% per year (95% CI: 0.66–1.68%), with no significant differences between species and the community-level medium. Once removed, only 7.2% (95% CI: 4.4–12.9%) were replaced annually. Presence of overhead utility lines influenced tree removal rates while age, diameter-at-breast-height, and prior tree condition influenced tree growth. Overall live aboveground biomass in sampled sites was 713.29 Mg in 2000 and increased to 877.36 Mg by 2014. Biomass gain from growth outweighed loss from removals nearly three-fold; replacement contributed 0.5% of the total biomass gain. We conclude that to increase the resilience of the street tree population will require 1) an increase in percent of full stocking or biomass stock and 2) a shift in the species palette to favor species less vulnerable to pests and expected disturbance from climate change and 3) ongoing monitoring to detect departures from baseline demographic rates.

## 1. Introduction

Quantifying the components of forest demography: recruitment, mortality and growth, is key to understanding future change and implications of different management regimes. The same principles apply to urban forest demography, except processes can vary. New trees, termed “recruitment” can occur from natural regeneration or plantings in new sites and old sites as “replacements”. In urban forests, the term “removal rate” is used in lieu of mortality rate because humans remove trees at any time and it is often difficult to know if the tree died of biological causes or was removed for other reasons. Growth of surviving trees is important to quantify because much of the urban forest's function and value is modeled on the size of its trees (McPherson et al., 2016b). It is the combination of growth, removal and replacement plantings that drives the changes in the structure and function of the population (Fahey et al., 2013), and therefore all three demographic components need to be assessed to gain a complete perspective of the state of the urban forest.

Stability in municipal forest structure translates to predictable and sustained levels of ecosystem services. Stability minimizes the risk of catastrophic losses that would inevitably lead to disruptions in

municipal budgets and ecosystem function (McPherson and Kotow, 2013). In the urban forestry context, stability has been defined as the low probability of incurring tree loss leading to disruptions in management and diminished flow of functional values and benefits from trees (Richards, 1983). Two metrics can be used to define stability in a street tree population. The first is continuity of stocking level or percent of full stocking, where full stocking is two trees per 15.2 m of street length (Wray and Prestemon, 1983). Stocking level reflects street tree density without reference to tree size. The second metric is continuity of biomass stock over time, where biomass is calculated for each tree at periodic intervals using allometric equations. Biomass incorporates differences in tree sizes and reflects the magnitude of the population's function and value. In terms of stocking level, stability occurs when replacement plantings match or exceed removals to the cohort under study. In terms of biomass stock, stability occurs when biomass from replacement plantings and growth of survivors matches or outpaces loss from removals. Just as the current urban forest is a reflection of the integrated outcome of past removals, growth, and plantings, the stability of the forest in the future is determined by the balance of these demographic components.

Previous work on demographic rates have taken a parsed approach.

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Most studies have solely looked at the survival or growth of newly planted trees (e.g., [Impens and Delcarte, 1979](#); [Ko et al., 2015](#); [Nowak et al., 1990](#); [Roman et al., 2014](#); [Thompson et al., 2004](#)) and replacement rates have not been explicitly reported in urban forestry literature. Fewer studies have considered the balance of inputs and outputs (but see [Roman et al., 2014](#)) and to our knowledge, not one has analyzed stability in terms of stocking or biomass. Ideally, demographic rates should be quantified on a species-specific level ([Roman et al., 2016](#)), but very little information exists for empirical tree growth rates. In light of the central role urban forests play in uptake of pollutants and carbon in cities ([Brack, 2002](#); [Escobedo et al., 2011](#); [Nowak and Crane, 2002](#); [Nowak et al., 2006](#)) it is important to build our understanding of species-level dynamics within the population and assess temporal patterns. In this study, we demonstrate application of the demographic approach successfully used in wildland forest systems (e.g., [Fahey et al., 2013](#); [Levine et al., 2016](#); [van Doorn et al., 2011](#)) to assess drivers of urban forest change.

Within the urban forest landscape, street tree populations differ from remnant forest patches or afforested parks due to being of planted origin and high management intensity ([Zipperer et al., 1997](#)). Although street trees are only a small part of the urban forest, they contribute considerable ecosystem services ([McPherson et al., 2016b](#)) as well as disservices to the urban landscape ([Escobedo et al., 2011](#); [Pataki et al., 2011](#)) and thus require municipal budget spending for their management. For example, in California, 9.1 million street trees make up approximately 5.2% of the 173.2 million trees in urban areas ([McPherson et al., 2017](#)) and contribute an estimated annual value of \$1 billion in ecosystem services ([McPherson et al., 2016b](#)). Evaluating drivers of change to street tree populations can help municipalities plan management strategies.

The structure of the urban forest is typically assessed from one-time measurements described as static “snapshots in time” ([McPherson and Kotow, 2013](#); e.g. municipal inventories, i-Tree Eco plot data, aerial imagery). These are the methodological building blocks for assessing changes in the urban forest, but unless individual trees are tracked and repeatedly measured over time, the relative contributions of removal, growth, and plantings to changes in population numbers and total biomass stock remains unknown.

Analysis of these drivers of change by way of longitudinal tracking of individual trees can be a valuable tool for predicting population stability and identifying trends. An initial estimate of removal, growth and replacement rates provides a baseline to which future measurements can be compared to evaluate changes in forest structure and value. Estimates of demographic rates form the basis of many projection tools and benefit calculators. Increased accuracy of growth and removal rates for a broader range of species will increase the usefulness of these tools. In addition, knowing how demographic rates differ by species allows managers with limited funding to identify the most vulnerable segments of the population and make more informed decisions about where to focus management efforts. For example, low removal rates can imply fewer opportunities to improve species and age diversity. Small changes in demographic rates can deeply affect future forest structure, composition and dynamics.

To help understand street tree dynamics in a maturing urban forest, we quantified species-level removal, growth, and replacement over a 14-year time interval in Claremont, CA. A sample of 762 street tree sites in Claremont, CA first measured in 2000 ([McPherson et al., 2016b](#); [Morani et al., 2011](#)) was revisited in 2014 to collect information on tree presence (e.g., removed, replaced), structural characteristics (e.g., diameter-at-breast-height [dbh], tree height), and physical factors surrounding the site (e.g., sidewalk damage). The objectives of this study were to (1) quantify species-level removal, growth, and replacement rates, (2) evaluate the contribution of these components to changes in biomass, (3) identify determinants of growth and removal in this all-aged population, and (4) qualitatively assess the trajectory of the street tree population given the observed demographic rates.

## 2. Methods

### 2.1. Study system

The study area, Claremont, CA is situated at the eastern end of Los Angeles County and is characterized by a Mediterranean climate. In the summer months, average temperature highs reach the 30 s (C), while the winters remain mild with average lows above freezing, and highs in the upper 10 s (C) ([Western Regional Climate Center, 2017](#)). The mean annual precipitation is 591 mm with most of it occurring in the winter. In fall months, Claremont can receive strong gusts of wind (i.e. “Santa Ana Winds”).

Claremont has a long urban forest tradition and has been awarded the Arbor Day Foundation’s Tree City USA award for over 20 years. Claremont has a population of 34,926 (2010 United States Census) and a land area of 34.6 km<sup>2</sup>. A 2011 municipal street tree inventory reported 19,980 street trees for tree density of 97 trees per street kilometer ([McPherson et al., 2016a](#)). In 2011 street tree sites were 74% stocked.

### 2.2. Data collection

In 2000, street trees from 21 of the most abundant species (in addition to a palm species we excluded from our study) were stratified by size class and randomly selected for measurement ([McPherson et al., 2016b](#)). The 21 species sampled in this study represent 63% of the total street tree population in 2000 ( $n = 23,554$ ) and 64% of the street tree population in 2011 ( $n = 19,903$ ). These 21 species captured 70% of the total street tree leaf area and 67% by importance value in 2000 (Appendix A in Supplementary material). A detailed description of methods of the original survey can be found in [McPherson et al. \(2016b\)](#). Claremont, CA was selected as a “reference city” representative of the Inland Empire climate zone ([Brenzel, 1997](#)). The sample was stratified into nine dbh classes (0–7.6, 7.6–15.2, 15.2–30.5, 30.5–45.7, 45.7–61.0, 61.0–76.2, 76.2–91.4, 91.4–106.7, and > 106.7 cm). The most recent municipal inventory at the time was used to determine the most abundant species. Although the trees were not tagged, location information was recorded in reference to the nearest residential address. Measurements included tree dimension metrics such as dbh (i.e. trunk diameter at 1.37 m), tree height, height to crown base, the average of two crown diameters, crown height and leaf area. In addition to tree dimensions, tree condition, management needs, and infrastructure were assessed (e.g., presence of utility lines and growing space). Tree age was obtained through a variety of methods including interviews with local residents and the city’s urban forester and historical records such as street and home construction dates, historical planting records, and aerial and historical photos.

In 2014, we revisited 752 of the original sites. “Sites” refer to locations of the trees measured in 2000—the same trees might not be present in the second census. The main goal of the resurvey was to account for live trees in the initial survey, identify replacements, measure dbh and assess health. Trees found at the sites were recorded to species and measured for dbh, tree height, crown height, and crown width. Sidewalk damage, growing space, presence of overhead utilities, and tree condition were also assessed. For multi-stem trees, the quadratic mean of the individual diameters was used as the measure of trunk diameter.

Of the 752 sites total that were surveyed in 2000, we were unable to find 20 sites in 2014 (3% of the total), leaving only 732 sites surveyed in 2014. They could not be located due to issues with site location as recorded in 2000. Site location information was based on nearest residence address and relative location within the address which made it difficult to distinguish between similar trees planted at a single residence. In one instance, there was a typo in the address listing; in another instance the reworked landscaping made it impossible to determine the exact location of the site. When there was uncertainty,

photographs from 2000 were used to identify the correct site. For the purposes of calculating demographic rates, the missing sites were excluded from further analysis.

Of the remaining 732 sites, trees in 10 sites were incorrectly identified due to lack of clarity in location description and thus a different tree from the original dataset was measured. The discrepancy was found through old and new photos as well as Google Street View imagery. The trees in these 10 sites as identified by imagery were counted as survivors from the original census after verification from imagery but were not used for growth analysis since they were not measured in the second census. As a result, a total of 722 individual trees were present in between 2000 and 2014 and all but one were evaluated for use in growth rate estimates. The one that was omitted was because it was standing dead and had not yet been removed, thus it was not used in the growth analysis but was tallied as not being removed.

### 2.3. Demography

The demography of Claremont's street trees was analyzed based on calculations of tree removal, growth and replacement rates. To estimate annual rates of removal, growth and replacement for the overall community and individual species levels, a hierarchical Bayesian model was used following Condit et al. (2006). This model incorporates both within-species and between-species variation by defining a joint probability distribution incorporating both the community and species-level distributions. This approach is particularly useful for estimating demographic rates for scant observations of rare species because the species level is informed by the overall community mean.

The annual removal rates (%yr<sup>-1</sup>) were calculated as per Condit et al. (2006):

$$\text{Annual removal rate} = \frac{\ln N_0 - \ln N_t}{t} \quad (1)$$

where  $N_0$  is the number of live trees in the first census,  $N_t$  is the number of trees in the second census, and  $t$  is the time between the first and second census. Removal rates were calculated for each species and the overall community.

For surviving trees with dbh measurements in each census period, average annual growth rates of survivors (G) was calculated following Condit et al. (2006) for the period between 2000 and 2014:

$$G_k = \frac{\ln(\text{dbh}_{k,j+1}) - \ln(\text{dbh}_{k,j})}{t} \quad (2)$$

where  $k$  is the individual tree,  $\text{dbh}$  is the tree diameter (cm),  $j$  is the census, and  $t$  is the time interval between  $j$  and  $j + 1$  calculated for each tree. Growth rates were calculated for each species and the overall community. Species-level growth is reported as relative growth rate (RGR, as the percentage relative to the size when it was first measured), which accounts for any size-related growth differences. For example, 7 cm of growth is a larger percentage for a 10 cm dbh tree than for a 30 cm dbh tree. Growth rates were later converted to cm · yr<sup>-1</sup> for the median dbh of each species to facilitate comparisons with other studies.

Growth estimates based on differences in dbh between censuses can be problematic due to variations in the location along the bole where the measurements are taken in consecutive censuses. Trees may have structural issues (i.e. splitting bole, peeling bark) that make it impossible to accurately measure the change in diameter. These variations may result in exceptionally large changes in dbh or negative growth rates which are biologically unrealistic. To deal with this issue, Condit et al. (2006) recommend excluding negative values from the growth rate calculations. Since this action may lead to bias in overestimating overall growth rates, we excluded an equal number of individuals at the upper growth rate distribution as defined by an interquartile range (iqr) (van Doorn et al., 2011). This nonparametric analytical approach is an unbiased statistical determination of outliers since it assesses both ends

of the distribution evenly (Sokal and Rohlf, 1981). First the growth data were ranked from lowest numerical value to highest. Outliers were removed using the interquartile range method, which uses the first quartile (q1), the third quartile (q3) and a multiple of the interquartile range of the ranked data to define outliers: observations outside of  $q1 - (1.5 \cdot \text{iqr})$  and  $q3 + (1.5 \cdot \text{iqr})$  were omitted from the growth analysis. This process eliminated at most 5% of survivors (33 out of 621). By species, the smallest percentage of excluded outliers was 0% (*Cinnamomum camphora*, 0 out of 34 survivors) and the largest was 16% (*Magnolia grandiflora*, 5 out of 31 survivors). The remaining negative growth rates of small magnitude not eliminated by the iqr method were recalculated by adding the overall minimum growth change (across species) to the 2014 census dbh (Condit et al., 2006). This was a necessary step because the log component of the relative growth equation (Eq. (1)) requires non-negative values. Replacement planting rates ( $k$ , %·yr<sup>-1</sup>) were calculated from direct observations of trees planted since the initial census at the sites where the original trees were removed. Replacement rates were calculated for each species and the overall community.

Removal, growth and replacement rates of the overall community and 21 tree species were estimated using a hierarchical Bayesian approach. Following Bayes theorem, the joint probability distribution for the overall community is defined as the product of the community-level and species-level probabilities; the distribution of each was based on observed data. For removal, the community level distribution followed an exponential distribution among the 21 species whereas at the species level it followed a binomial distribution. The model has the following form:

$$S_j | \theta_j, N_j \sim \text{Binomial}(\theta_j, N_j),$$

$$\theta_j = \exp(-14m_j),$$

$$m_j | \lambda \sim \text{Exponential}(\lambda),$$

$$\lambda \sim \text{Uniform}(0, 1000)$$

where  $S_j$  is the response variable for survivor observations of species  $j$  ( $j = 1, 2, \dots, 21$ ) which follows a distribution from the binomial family where  $\theta_j$  is the removal probability annualized across 14 years, and  $N_j$  is the number of trials.  $m_j$  is the rate constant of removal which follows an exponential distribution with rate  $\lambda$ .

For relative growth, both the community and species level distributions followed a log-normal distribution. The model has the following form:

$$\text{rate}_{ij} | g_j, \tau_1 \sim \text{Log-normal}(g_j, \tau_1),$$

$$g_j | \sigma, \tau_2 \sim \text{Log-normal}(\sigma, \tau_2),$$

$$\sigma \sim \text{Uniform}(-1000, 1000),$$

$$\tau_1 \sim \text{Gamma}(0.001, 0.001),$$

$$\tau_2 \sim \text{Gamma}(0.001, 0.001)$$

where  $\text{rate}_{ij}$  is the response variable for growth observation  $i$  ( $i = 1, 2, \dots, N$ ) of species  $j$  ( $j = 1, 2, \dots, 21$ ), which follows a distribution from the log-normal family with mean  $g_j$  and dispersion parameter  $\tau_1$ .  $g_j$  is the community-level response variable for species' growth  $j$  which follows a log-normal distribution with mean  $\sigma$  and dispersion parameter  $\tau_2$ .

For replacement plantings, the community level distribution was most closely approximated by an exponential distribution while the species level followed a binomial distribution. The model has the following form:

$$V_j | \mu_j, N_j \sim \text{Binomial}(\mu_j, N_j),$$

$$\mu_j = \exp(-14r_j),$$

$$r_j | \delta \sim \text{Exponential}(\delta),$$

$\delta \sim \text{Uniform}(0, 1000)$

where  $V_j$  is the response variable for vacancy (lack of replacements) observations of species  $j$  ( $j = 1, 2, \dots, 21$ ) which follows a distribution from the binomial family with  $\mu_j$  being the replacement probability annualized over 14 years, and  $N_j$  being the number of trials.  $r_j$  is the rate constant of replacement which follows an exponential distribution with rate  $\delta$ .

The Markov chain Monte Carlo technique was used to numerically solve the integration and to fit the parameters. We ran the chain for 10,000 iterations with a 1000 run burn-in. We report median values and the 2.5th and 97.5th percentiles of the stored chains (i.e., credibility intervals; Clark, 2005) and define a significant difference between species or inventory intervals to be non-overlapping 95% Bayesian credible intervals (CI). The analysis was implemented in software packages R statistical software version 3.2.2 (R Core Team, 2012) and OpenBUGS 3.2.3 (Lunn et al., 2000).

## 2.4. Biomass

Demographic rates on their own lack information on the magnitude of change in relation to biomass and carbon storage. Gains and losses in aboveground biomass by species were estimated for the three different components: removal, growth, and replacement. First, aboveground biomass was calculated for all trees in the original survey using allometric equations (McPherson et al., 2016b). Biomass from trees removed by 2014 was added to the “loss from removal” category. Biomass from trees that survived until 2014 was added to the “gain from growth” category. Similarly, biomass from trees that had been replaced counted towards the “gain from replacement” category.

## 2.5. Determinants of tree removal

We used conditional inference trees to identify the most important factors contributing to the 14-year tree survival outcome. The ‘party’ package in R (Hothorn et al., 2006) provides nonparametric classification trees via the function ‘ctree’. We expected tree removal to be influenced by tree size (dbh), presence of overhead utility lines, sidewalk damage, tree condition (in 2000), and availability of growing space (Appendix B in Supplementary material).

Next, to assess the direction of the relationship between the risk factor(s) selected by the conditional reference tree and 14-year survival outcome, we used a binary logistic regression model with random effects (Hosmer et al., 2013). We fit the model using the function ‘glmer’ in R package ‘lme4’ (Bates et al., 2015). Species identity was treated as a random term in the model. Logistic regression was used to predict determinants of survival since survival is a binary outcome variable (removed/survived).

## 2.6. Determinants of tree growth

As with the tree removal analysis, we used conditional inference trees to identify the most important factors contributing to tree relative growth rate (Appendix B in Supplementary material). We expected tree relative growth rate to increase as a function of tree size (dbh) and age, and decrease as a function of poor tree condition (in original survey year), growing space, and presence of overhead utility lines due to pruning. The classification model did not depend on presence of utility and growing space as factors in determining tree growth and thus were removed from further consideration (Appendix B in Supplementary material).

To assess the relationship between selected predictor variables and 14-year tree relative growth rates, we initially used linear mixed models (using the ‘lmer’ function in R package ‘lme4’ (Bates et al., 2015)) which proved to be an unsuitable fit since there was a discernable pattern in the relationship of residuals to fitted estimates (Pinheiro and Bates, 2000). Further diagnostics showed that the relationship of age and dbh to relative growth rate (RGR) could be non-linear. Therefore, to assess the relationship between selected predictor variables and relative growth rates we used a generalized additive model (GAM). Using the package ‘mgcv’ in R (Wood, 2017), RGR was modeled as a function of age in 2014, dbh, and condition in 2000. Species identity was treated as a random term in the model.

Our metric for growth was relative growth rate (RGR), calculated as  $\log(\text{dbh in 2014}) - \log(\text{dbh in 2000})$  over a 14-year period.

Following are the specifics of the fitted regressive model:

$$Y_{ij} = \beta_0 + t2(\text{Age}_{14ij}) + s(\text{DBH}_{ij}) + s(\text{Cond}_{ij}) + \text{SpCode}_j + \varepsilon_{ij} \quad (3)$$

where  $y_{ij}$  is the RGR of the  $i$ th tree of the  $j$ th species =  $\log(\text{dbh}_{2014}) - \log(\text{dbh}_{2000})$ ;

$\beta_0$  is the intercept of the regression model;

$t2(\text{Age}_{14})$  is a tensor product smooth function of the age of the tree in 2014;

$s(\text{DBH})$ ,  $s(\text{Cond})$  are smooth spline functions of the DBH and tree condition in 2000;

$\text{SpCode}$  is the random intercept for species identity of the  $i$ th tree; and

$\varepsilon_i$  is the normally and independently distributed error with mean 0 and variance  $\sigma^2$ .

## 3. Results

### 3.1. Overall demography

Of the 732 sites visited in both 2000 and 2014, 632 sites had survivors and 100 sites had removed trees (Fig. 1). 621 of the 632 survivors were measured both years and were the basis for estimating

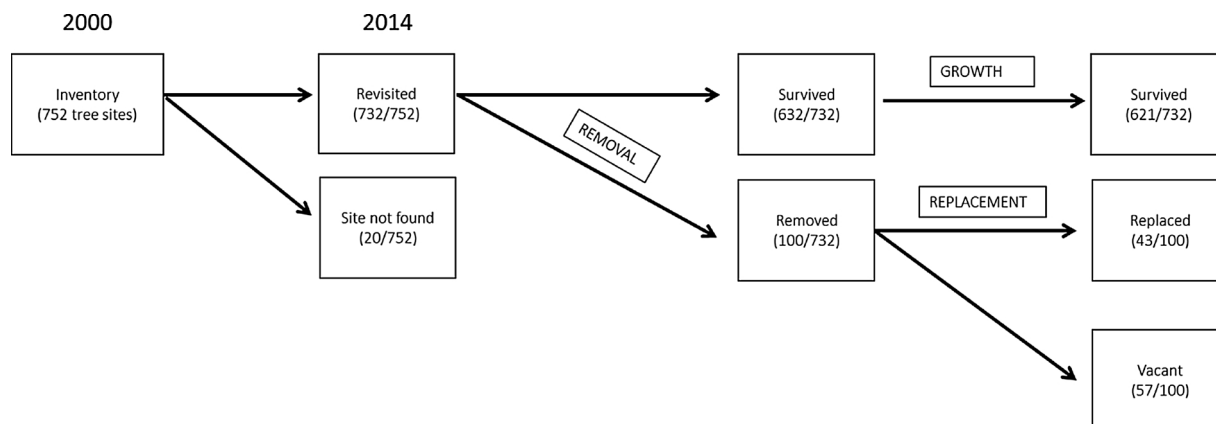


Fig. 1. The fate of inventoried trees between 2000 and 2014. Demographic components are outlined.



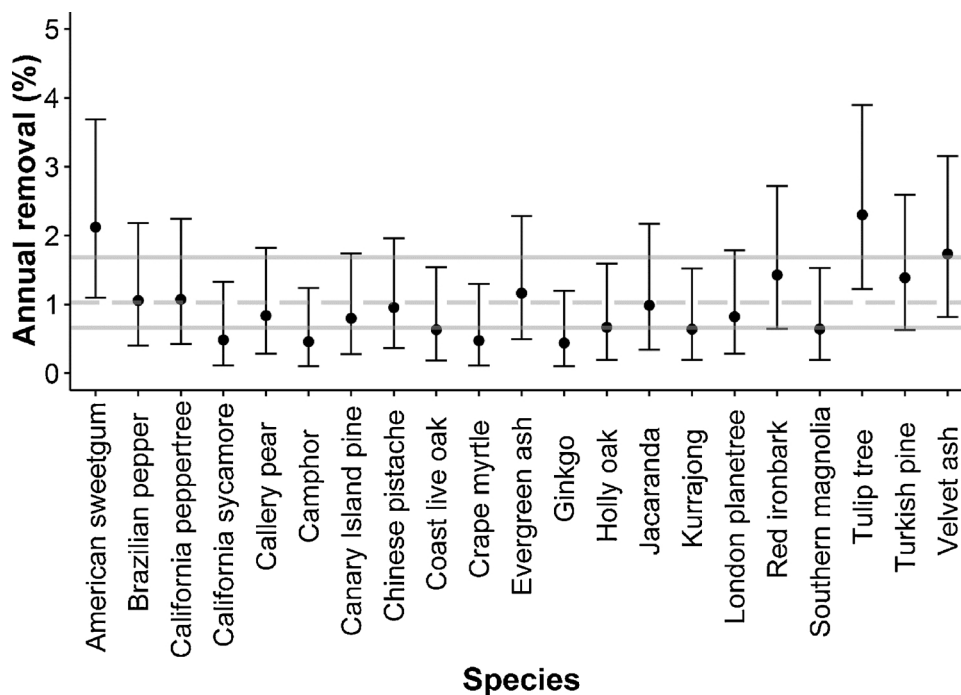


Fig. 2. Median annual removal rates (% · yr<sup>-1</sup>) and 95% credibility intervals for 21 street tree species in Claremont, CA. The community-level median is marked by a dashed grey line with corresponding 95% credibility intervals.

growth rates (see methods for outlier omissions). Only 43 of these 100 sites with removed trees were replanted (of which 1 replacement tree died by 2014). The rest of the 57 sites did not receive a replacement tree.

### 3.2. Removal

Taking all 21 species into account, the community-level median removal rate was 1.03% per year (95% CI: 0.66–1.68%; Fig. 2). *Liriodendron tulipifera* (tulip tree) had the highest removal rate at 2.30% per year (95% CI: 1.22–3.90%) while *Ginkgo biloba* (ginkgo) had the lowest at 0.44% per year (95% CI: 0.10–1.20%). Individual species' removal rates did not differ from the community-level median removal rate.

To identify determinants of removals we constructed a conditional inference tree model with the following covariates: dbh in 2000, tree height in 2000, condition in 2000, evidence of sidewalk damage in 2000, and presence of utility lines in 2014. The covariate with the largest association to tree removal was presence of overhead utility lines, significant at the  $p < 0.05$  level ( $p = 0.034$ ) (Appendix B in Supplementary material). Sites with utility lines ( $n = 68$ ) had a significantly higher tree removal rate compared to sites with no utility lines present ( $n = 664$ ). There was no significant association between any of the other covariates and the response variable.

To assess the direction of the relationship between the risk factor selected by the conditional inference tree and 14-year survival outcome, we used a mixed effects logistic regression model. Results from the model using utility lines as a fixed term and species as a random term showed that presence of utility lines had a negative effect on tree removal (Table 1). The ratio of odds for trees under utility lines to the odds for trees without utility lines overhead was 0.156, meaning trees under utility lines were less likely to be removed.

### 3.3. Growth

The community-level (all species) median RGR was 1.41% per year (95% CI: 1.21–1.65%; Fig. 3). The only species growing at a significantly higher rate than the community-level median were *Pinus brutia* (Turkish pine) at 2.44% per year (95% CI: 1.70–3.52%) or 0.7 cm

per year for a 26.6 cm dbh pine and *Pistacia chinensis* (Chinese pistache) at 2.43% per year (95% CI: 1.74–3.45%) or 1.6 cm per year for a 66.8 cm dbh pistache. Slower growing species included *Fraxinus uhdei* (evergreen ash), *Cinnamomum camphora* (camphor), *Brachychiton populneum* (kurrajong), and *Lagerstroemia indica* (crape myrtle). Crape myrtle grew at 0.91% annual RGR (95% CI: 0.63–1.28%) or 0.2 cm per year for a 20 cm dbh crape myrtle.

Determinants of growth were analyzed using a generalized additive model. Covariates showing associations to relative growth (over 14 years) were age in 2014, dbh, and condition in 2000 at the  $p < 0.001$  level (Table 2). Since RGR is calculated by subtracting the log of dbh in different years, the scale of the linear predictor in the resulting model is on the log scale. The generalized additive model explained 77.9% deviance and the adjusted  $R^2$  was 0.765.

The estimated mean growth per year and corresponding standard error from the fitted model was 10.15% (se: 7.410). Relative growth was highest at younger tree age but dropped dramatically by year 30, followed by a flat or slightly decreasing pattern (Fig. 4). A similar pattern existed for initial dbh, with smaller diameter trees exhibiting higher growth rates that quickly dropped off and flattened. Relative growth also varied with condition of the trees at the initial survey. Poor condition ( $< 60$  on a scale from 1 to 100) had a negative effect on growth. For trees with  $> 60$  condition rating, growth rates increased and then stabilized as condition improved.

### 3.4. Replacement

The community-level annualized median replacement rate of trees removed in Claremont between 2000 and 2014 was 7.2% [95% CI: 4.4–12.9%] (Fig. 5). No individual species were significantly different from the community-level median. *Jacaranda mimosifolia* (jacaranda) was the least replaced species at 1.0% [95% CI: 0.04–5.3%] while kurrajong [95% CI: 3.6–40.8%] and *magnolia grandiflora* (southern magnolia) [95% CI: 3.5–40.5%] had the highest median replacement rates at 13.2%.

Of the trees removed between 2000 and 2014, 50% were replaced with the same species as had existed at the sites in 2000 while the other 50% were replaced with species new to the tree sites (but not necessarily new to the inventory; Fig. 6). Tulip tree, the species with the

**Table 1**  
Effect of utility line presence on tree removal from a mixed effects logistic regression model fit by maximum likelihood. Results describe how utility line presence affects occurrence of tree removal. Species identity of the observed trees was included as a random term in the analysis, while presence/absence of utility lines was a fixed term. A total of 732 tree observations were analyzed in this model.

	Parameter estimate	Std. error	Odds ratio	95% confidence interval of the odds ratio	z-value	p-value	
(Intercept)	−1.802	0.146	0.165	[0.120, 0.218]	−12.324	< 2e-16	***
Utility (present)	−1.833	0.735	0.156	[0.026, 0.533]	−2.495	0.0126	*

Note: Std.error = standard error of the mean.  
\* Represents significance at  $p < 0.05$  level.  
\*\*\* Represents significance at  $p < 0.001$  level.

highest removal rates, was exclusively used as a replacement of removed tulip trees. Ginkgo was the second most commonly used replacement species, and most of the time it replaced a different species. Nearly half of the removed trees (47%) were in the semi-mature size class (30–61 cm dbh), 28% were in the maturing class (15–30 cm dbh), 13% were in the mature class (> 61 cm dbh) and 12% were in the smallest class (< 15 cm dbh). The size category with the highest replacement rate was the mature class (60%) followed by the juvenile class (50%) and semi-mature class (43%); the lowest rate was in the maturing class (34%).

3.5. Stocking

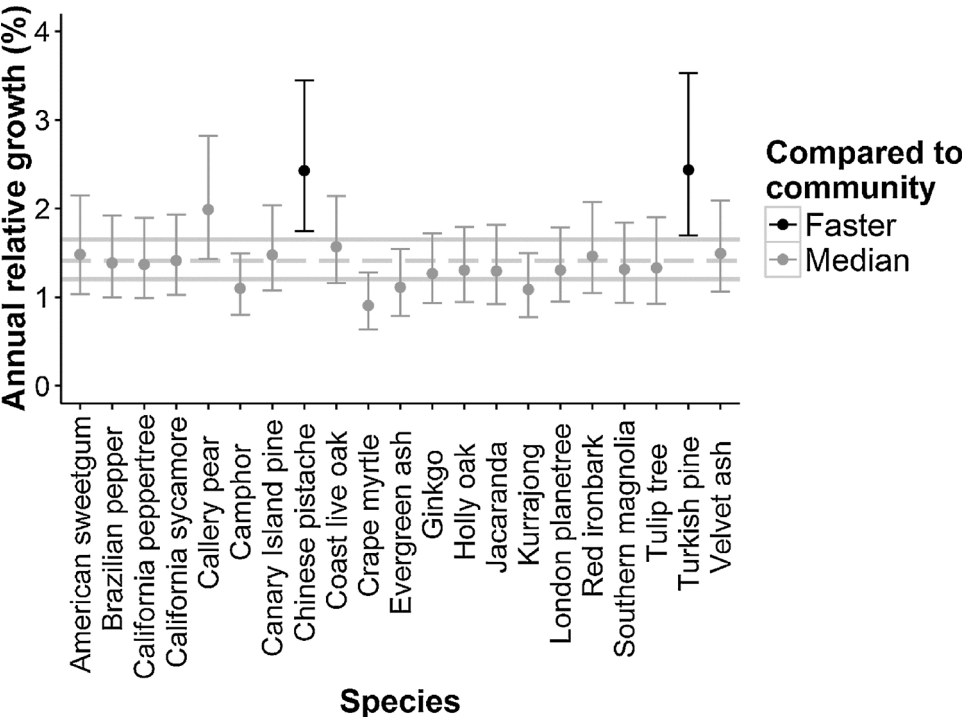
Stocking levels were calculated using two metrics. The first metric was change in abundance based on draw down rates from removal and replacement. A simple simulation of stocking level change for the cohort of sites sampled was inferred to Claremont’s street tree population based on an initial stocking in 2011 of 74% (West Coast Arborists 2011, unpublished data). Although the replacement planting rates in Claremont were low, the low removal rates resulted in an overall minimal decrease in stocking. Given a 1.03% community-level removal rate and 7.3% replacement rate (on removed trees; for a total of 0.0748% reduction in stocking annually), Claremont’s projected stocking after 20 years would be 73%, assuming no new sites had been added, or no sites lost. Variation among species was slight, ranging from 71.0% stocking after 20 years if all trees were American

sweetgums (*Liquidambar styraciflua*) to 73.9% if all trees were crape myrtles. These species-level simulations assume that all species start at 74% stocking because we did not have the species-level stocking data. The second metric used to measure stocking was change in live aboveground biomass between 2000 and 2014. Overall, aboveground biomass in sampled sites increased from 713.29 Mg in 2000 to 877.36 Mg in 2014. Biomass gain from growth outweighed loss from removals nearly three-fold (Table 3). Replacement only contributed 0.5% of the total biomass gain.

There were differences in species-level contributions to net biomass change. Five species, namely *Quercus agrifolia* (coast live oak), Canary Island pine, Turkish pine, *Quercus ilex* (holly oak) and *Schinus molle* (California peppertree), accounted for 52% of the net change in biomass, driven by low loss from removals and large gain from growth (Table 3). The only species that had a net negative biomass change was *Fraxinus velutina* (velvet ash) due to loss from removals exceeding gains from growth.

The largest amount of biomass lost from removals was in red ironbark, evergreen ash, and tulip tree, but growth from survivors more than made up for loss from removals.

Linking the biomass results to the demographic rates, the highest removal rates were in tulip tree, American sweetgum, velvet ash, red ironbark, Turkish pine at  $> 1.3\% \cdot \text{yr}^{-1}$ , but high rates did not necessarily result in high biomass losses. This is due to species differences in mature size and wood density. Most of the removed red ironbark and tulip tree were large-stature trees and their loss resulted in a



**Fig. 3.** Median annual relative growth rates ( $\% \cdot \text{yr}^{-1}$ ) and 95% credibility intervals for 21 street tree species in Claremont, CA. The community-level median is marked by a dashed line with corresponding 95% credibility intervals. Black dots represent species growing significantly faster than the community-level median.

**Table 2**

Results of a generalized additive model with covariates age, dbh and condition and tests of association to relative growth rate data.

(Intercept)	Parameter estimate	Std. error	t-value	p-value	
	0.283558	0.005904	48.03	< 2e-16	***
	Edf	Ref.df	F-value	p-value	
(Age14)	3.873	3.985	0.156	0.0126	***
s(DBH)	8.623	8.957	16.645	< 2e-16	***
s(Cond)	5.348	6.463	4.399	0.000133	***

Note: Std.error = standard error of the mean.

\*\*\* Represents significance at  $p < 0.001$  level; Edf = estimated degrees of freedom for each of the smooths.

proportionally higher drop in biomass.

## 4. Discussion

### 4.1. Removal

Tulip tree and American sweetgum had the highest median removal rates but they were not statistically higher than other species. That sweetgum is associated with increased mortality is not surprising consider that substantial dieback has been reported due to Bacterial Leaf Scorch caused by the pathogen *Xylella fastidiosa* (J. Cawn, personal communication, Nov. 10, 2015). Tulip tree is particularly sensitive to drought because it has shallow roots and in the wild would typically be found in moist areas. The City of Claremont has responded to concerns about the suitability of these species: the Designated Street Tree List (City of Claremont Designated Street Tree List, 2016) shows no tulip tree or American sweetgum.

California's unprecedented droughts during the past decade (California Department of Water Resources, 2016) are likely to have exposed many street trees to water stress, a critical factor in urban tree survival (Whitlow and Bassuk, 1987). During 2012–2016, emergency restrictions on irrigation were implemented but trees were not exempted (California Department of Water Resources, 2015). Water stress may be contributing to higher removal rates of drought intolerant

species native to temperate climates (i.e., sweetgum, ash, pear). For example, Koeser et al. (2014) tracked trees 2–5 years after planting in Florida and found that on-site irrigation was a big factor in tree survival and growth (especially for magnolia).

The overall annual removal rate in Claremont was calculated for an all-aged cohort of trees over more than a decade. Such information is rare. In particular, many studies have focused on establishment success/mortality of newly planted trees (e.g., Impens and Delcarte, 1979; Nowak et al., 1990; Vogt et al., 2015). The annual removal rate in Claremont (median: 1.03%; 95% CI: 0.66–1.68%) was lower than the range calculated from a meta-analysis of 16 street tree survival studies where the annual survival rate was 94.9–96.5% and the corresponding annual mortality rate was 3.5–5.1% (Roman and Scatena, 2011). The only comparable study we know of that followed one cohort of street trees of all ages was conducted in Milwaukee, WI (Koeser et al., 2013). Survival was measured over a 10-year period; all species together had 83.6% survival. No species overlapped with the Claremont study, but two species common to genera studied in Claremont were present: *Fraxinus americana* (79.2%) and *Fraxinus pennsylvanica* (88.4%). In Claremont, 84% of the original *F. uhdei* and 76% of *F. velutina* had survived between 2000 and 2014 (both  $< 2\% \cdot \text{yr}^{-1}$  median removal rate; Fig. 2), within the range reported by Koeser et al. (2013).

In evaluating removal rates, one cause for concern in terms of effect on population projections would be if most of the removals were

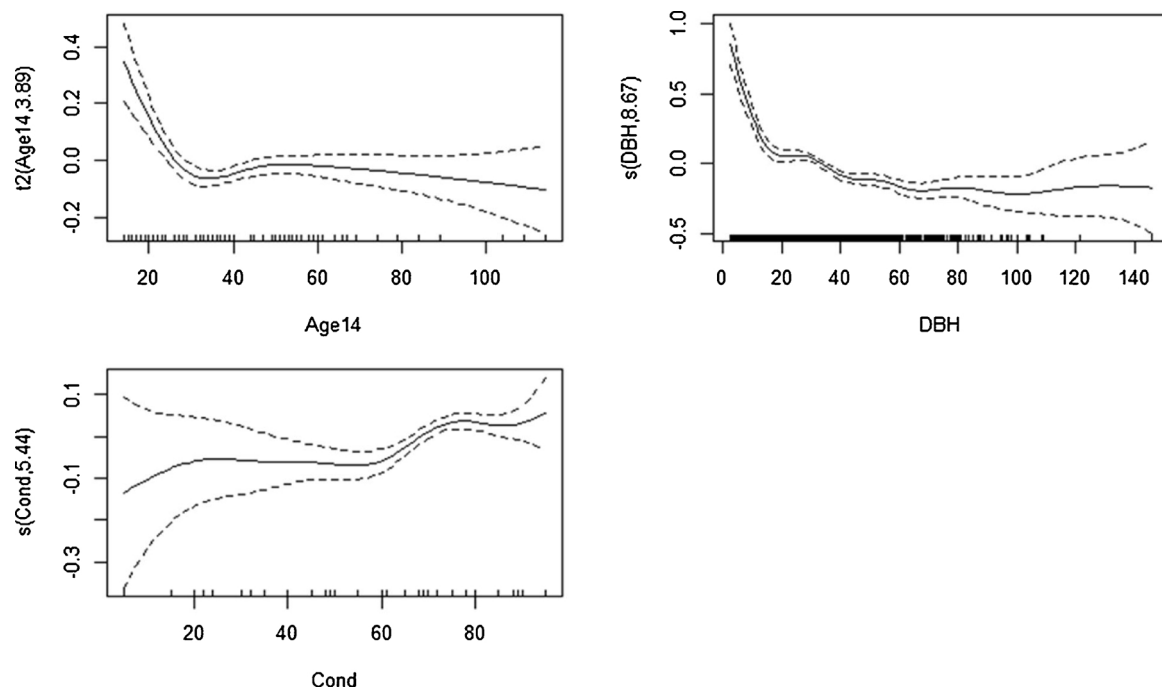


Fig. 4. Estimated effects (smooth spline functions from Eq. (3)) of age in 2014, dbh, and tree condition (in 2000) on relative growth rate. The linear predictor (relative growth rate over 14 years) is on the log scale.

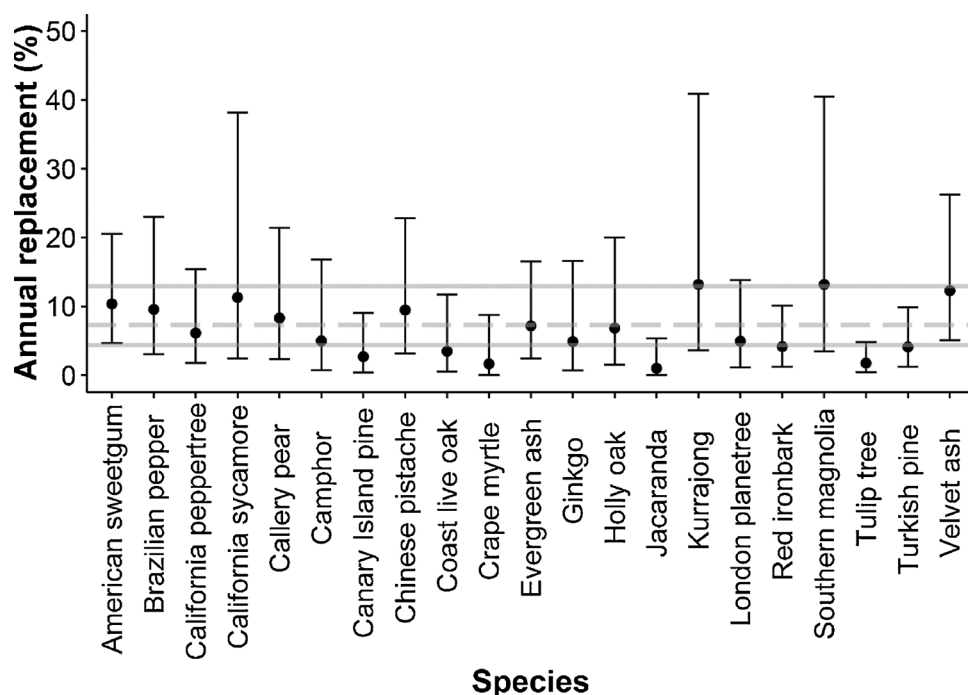


Fig. 5. Median annual replacement rates ( $\% \cdot \text{yr}^{-1}$ ) and 95% credibility intervals for 21 street tree species in Claremont, CA. The community-level median is marked by a dashed line with corresponding 95% credibility intervals.

occurring in the establishment period (years 1–2). This does not appear to be the case in Claremont. Of the 100 trees that were removed over the 14 year period, only 2 were 1–2 years old in 2000 and we do not know in which year they were removed. At the earliest, they were removed sometime in 2000 after the survey, but at the latest they could have been removed in 2014 not long before our survey. Low removal rates for young trees ( $< 4\%$  annually) in Claremont help ensure that trees will be present to provide the maximum amount of benefits possible over their lifetime (Widney et al., 2016).

Presence of overhead utility lines appeared to be the only factor important for long-term survivorship across the urban landscape. The direction of this trend was counterintuitive: utility line presence resulted in decreased tree removal. No other studies that we know of have looked at the effect of utility lines on removal. One possible reason for this result is that trees under utility lines receive more frequent pruning and care because of potential consequences on safety.

In our study, tree dbh, age, and prior condition did not play a major role in predicting removal. These findings contradict results from studies that have shown tree age, diameter and condition to be related to street tree mortality. A study of street tree survival in Milwaukee, WI, found that tree mortality increased as trunk diameter increased, as planting space width decreased, and as condition deteriorated (Koeser et al., 2013). It is surprising that poor prior condition of trees in 2000 did not result in higher tree removals. Based on Manion's (1991) "Decline-Disease Spiral" model, trees that are predisposed or exposed to stress are less likely to survive additional impacts from external or internal stresses. Richards (1979) found that mortality rates increase in large trees and in trees with declining health and increased crown dieback. Another study showed that street tree mortality rates increased with deteriorating crown condition and tree size (Nowak, 1986). Ko et al. (2015) found that mature tree size affected long-term survivorship, with small stature trees exhibiting higher mortality than medium

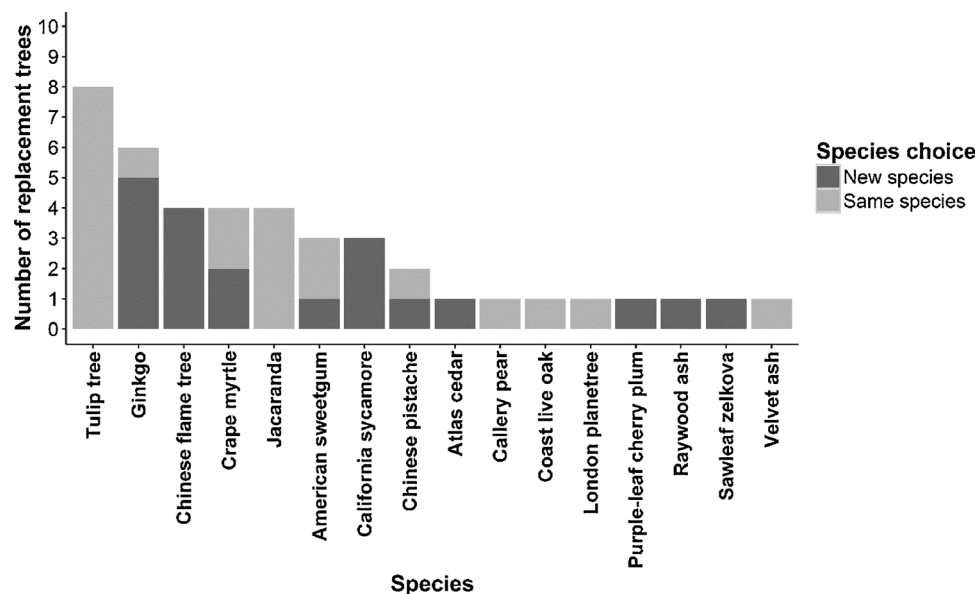


Fig. 6. Number of replacement trees by species. Species different from the original species planted at the site are marked in dark gray.



**Table 3**

The aboveground biomass (Mg) contributed by each demographic process in the inventory period 2000–2014.

Species	Loss from removal	Gain from replacement	Gain from growth	Net biomass change
American sweetgum, <i>Liquidambar styraciflua</i>	5.09	0.03	6.74	1.67
Atlas cedar, <i>Cedrus atlantica</i> 'Glaucous'	–	0.10	–	0.10
Brazilian pepper, <i>Schinus terebinthifolius</i>	2.89	0.00	9.79	6.90
California pepper tree, <i>Schinus molle</i>	7.87	0.00	19.08	11.21
California sycamore, <i>Platanus racemosa</i>	1.48	0.00	5.45	3.96
Callery pear, <i>Pyrus calleryana</i>	2.40	0.01	7.78	5.39
Canary Island pine, <i>Pinus canariensis</i>	1.93	0.00	25.63	23.71
camphor, <i>Cinnamomum camphora</i>	1.26	0.00	9.99	8.73
Chinese flame tree, <i>Koelreuteria bipinnata</i>	–	0.37	–	0.37
Chinese pistache, <i>Pistacia chinensis</i>	1.34	0.05	4.06	2.76
coast live oak, <i>Quercus agrifolia</i>	3.37	0.04	32.98	29.65
crape myrtle, <i>Lagerstroemia indica</i>	0.03	0.06	1.33	1.36
evergreen ash, <i>Fraxinus uhdei</i>	17.76	0.00	28.87	11.12
holly oak, <i>Quercus ilex</i>	2.49	0.00	15.29	12.80
ginkgo, <i>Ginkgo biloba</i>	0.29	0.26	6.95	6.91
red ironbark, <i>Eucalyptus sideroxylon</i>	21.74	0.00	25.58	3.83
jacaranda, <i>Jacaranda mimosifolia</i>	0.47	0.02	5.33	4.88
kurrajong, <i>Brachychiton populneus</i>	0.57	0.00	1.28	0.71
London planetree, <i>Platanus x acerifolia</i>	2.15	0.15	11.63	9.63
purple-leaf cherry plum, <i>Prunus cerasifera</i>	–	0.01	–	0.01
Raywood ash, <i>Fraxinus oxycarpa</i> 'Raywood'	–	0.02	–	0.02
sawleaf zelkova, <i>Zelkova serrata</i>	–	0.02	–	0.02
southern magnolia, <i>Magnolia grandiflora</i>	0.59	0.00	1.61	1.01
tulip tree, <i>Liriodendron tulipifera</i>	11.05	0.14	11.86	0.95
Turkish pine, <i>Pinus brutia</i>	3.95	0.00	22.20	18.25
velvet ash, <i>Fraxinus velutina</i>	3.13	0.03	0.97	–2.13
ALL	91.86	1.30	254.39	163.83

stature trees. We did not see such differences in our sample.

It was surprising that growing space was not a determinant of either growth or mortality. In our dataset, nearly 60% of trees were located in the least restricted category of growing space, typically front yard lawns with trees planted near the sidewalk. Trees located in more restricted spaces were not more likely to be removed. A possible explanation is that the area's deep alluvial soils allow tree roots to grow deeper than normal, so tree health is seldom compromised by a scarcity of nutrients, oxygen, and water. Past work has shown that at least in younger trees, trees with unrestricted growing space (which assumes more soil area), grew more than trees with less space (Berrang et al., 1985; Flückiger and Braun, 1999; Grabosky and Gilman, 2004; Kjellgren and Clark, 1992; Vrecenak et al., 1989). Long-term monitoring of street trees suggests that tree condition and mortality can be explained by growing space, among other factors (Hauer et al., 1994; Koeser et al., 2013).

Except for the positive effect of utility lines on survival, our analysis of determinants of removal would suggest that the physical limitations are not at play here. However, our analysis did not include factors such as construction related mechanical damage, differences in soil moisture, soil compaction, and soil temperature. Another limitation is that we did not have the monitoring data available to determine what if any social-economic factors might be playing a role in Claremont. Social-ecological factors have been found to be important drivers of change in establishment phase survivorship in urban ecosystems (Vogt et al., 2015). How much these factors play a role later in a tree's life remains to be seen. Long-term monitoring can shed light on this question.

#### 4.2. Growth

Two species in particular grew faster than the community-level median growth rate: Turkish pine and Chinese pistache. In terms of stored biomass though, they contributed in different capacities, with Turkish pine accumulating five times more biomass due to growth than the pistache (Table 3). In our sample, the dbh range of Turkish pine in 2000 was 3.8 cm to 77.3 cm while Chinese pistache was narrower at 2.9 to 30.5 cm, although both sample populations ranged in age from newly planted to 30 years old. The mature stature of the species differs with

Turkish pine growing to twice the height (20–25 m) of pistache (Urban Forest Ecosystems Institute, 2017). On a city-level though, the number of trees is important to note. In Claremont, Turkish pine represents less than 2% of the total population of street trees (ranking #20 based on counts alone) while pistache represents 4% (ranking #9). If benefits from biomass were of primary interest, planting Turkish pine would be a better planting choice than pistache because of the higher biomass yield per tree.

Species that did not grow significantly faster than the community-level median but contributed the biggest gains in biomass were coast live oak, evergreen ash, red ironbark, and Canary Island pine (Table 3). Together these four species made up 12% of the abundance in Claremont in 2011, and 16% of the importance value, with coast live oak being the most abundant and important (Appendix A in Supplementary material). However, due to species characteristics and relative size, these rankings do not necessarily match the benefits provided to the ecosystem and humans. Average annual benefits per tree are as follows: red ironbark: \$127.68, evergreen ash: \$368.95, coast live oak: \$55.81, Canary Island pine: \$308.26 (McPherson et al., 2016a). Coast live oak and red ironbark provide less monetary value because of their negative net effects on air quality whereas evergreen ash contributes a positive net effect on air quality and Canary Island pine nearly breaks even on air quality.

Although no species was growing significantly slower than the community-level median, crape myrtle exhibited the lowest median rate (Fig. 3). Crape myrtle was the most numerous tree in Claremont, comprising 9.5% of the total street tree population. Because it is slow growing and small statured, crape myrtle produces an average benefit value of only \$67.05/tree (McPherson et al., 2016a). However, this per-tree value is higher than for the faster growing coast live oak (\$56.00/tree) (Appendix A in Supplementary material). This highlights the need to have a variety of well-adapted species for planting when objectives (e.g., carbon storage, rainfall interception, aesthetics, heat island mitigation) and growing space (e.g., front yards, narrow strips, medians, tree wells in pavement) vary. Each species will have its tradeoffs. In cases where sites are too small to suit larger-stature trees, crape myrtles can excel.

In this study we considered the influence of tree age, dbh, condition, presence of utility lines, and growing space on tree growth. The amount of available space for the tree to grow has been known to influence the maximum diameter reached by a tree (Sanders et al., 2013). Surprisingly, growing space did not appear to be a factor determining growth in Claremont (Appendix B in Supplementary material). Our sample covered the spectrum of large trees in small growing spaces to small trees in large growing spaces. Other literature concerning determinants of growth has only looked at young tree establishment (Koeser et al., 2014). One limitation of our data collection is that our categories for growing space did not distinguish between surface cover types. For example, a tree growing in a large median with seemingly no space restriction would be in the same category as a tree growing in an open lawn. A stricter classification of growing space description could be beneficial in future work.

#### 4.3. Replacement

Replacement rates in Claremont over the period 2000–2014 were seemingly low, but there are no comparable studies with documented replacement rates. Judicious selection of replacement trees is one way to shape the future species mix and overall stability of the urban forest. One observation gleaned from this research is that the species selection used for replacements in Claremont is tending towards keeping the same species mix as before, with only a few new species. Replacing with the same species, if that species is relatively abundant and vulnerable to future climate stressors such as heat and drought, could have a destabilizing effect on the forest of the future. Such may be the case with tulip tree, *Pyrus calleryana* (Callery pear), American sweetgum, *Platanus x hispanica* (London planetree), and ash.

An important and immediate concern in the Southern California region is the susceptibility of species to major pests and diseases such as the Polyphagous Shot Hole Borer (*Euwallacea* sp.) which forms a symbiosis with a complex of fungal species *Fusarium euwallaceae*, *Graphium euwallaceae* and *Paracremonium pembeum* causing *Fusarium* Dieback disease (Eskalen et al., 2013). Planting underutilized species as replacements is a step in the right direction with regards to increasing diversity. For example, the four replacement species new to the cohort we studied (*Cedrus atlantica* ‘Glauc’, *Koeleruteria bipinnata*, *Prunus cerasifera*, and *Zelkova serrata*) each had fewer than 100 individuals in the 2011 inventory. An initial pest vulnerability assessment indicates that with the exception of the *P. cerasifera*, these species are relatively pest resistant. A more comprehensive assessment for a larger variety of replacement species would help inform whether they are likely to be successful under future predicted conditions.

#### 4.4. Stability

Our sample of Claremont’s street tree population is stable, both in terms of stocking level and biomass stock. This stable condition is currently driven by the low overall removal and replacement rates. With the low rates, Claremont’s street tree population will change very little in the near future. The newly planted trees take time to mature and although they may grow fast, the resulting biomass gain is less than it is for slower growth of larger trees. However, we expect a large flux in biomass stock once the larger trees senesce. This is because the 2011 inventory indicates that the bulk of trees are already mature: 26% juvenile, 28% semi-mature, 37% mature, and 8.3% senescent. This age structure suggests that a continued deficit of replacements will threaten long-term stability in stocking and biomass levels, as well as resultant canopy cover. In addition, future costs for intensive care and removal of senescent trees are likely to increase.

The City of Claremont has several strategies it can employ to reduce the long-term threats to the stability of its municipal forest. It can gradually increase stocking by planting trees in vacant sites as well as replacing each removed street tree. By establishing a stocking target,

such as 100% full stocking in 20 years, the total number of sites that need to contain trees can be calculated. For Claremont, this means increasing the number of filled tree sites from 19,980 in 2011 (74% full stocking) to 27,034 (100%) in 2030. This target can be reached in 20 years assuming the annual removal rate remains at 1.03%, each removed tree is replaced, and 150 trees are planted in vacant sites each year. In this projection, annual removal rates ranged from 206 to 282 and averaged 243 trees per year, while planting rates ranged from 356 to 432 trees per year, and averaged 393.

Planting a greater diversity of tree species that will be well adapted to future climate stressors is a second strategy that supports the full stocking goal. The number of trees to plant to achieve full stocking is highly sensitive to the removal rate, so selecting species likely to thrive is critical to controlling removal and replacement rates. Examples include natives (i.e., *Lyonothamnus floribundus* and *Prunus ilicifolia*) and species proven to perform in more arid urban areas (i.e., *Chilopsis linearis* and *Corymbia papuana*).

A third strategy is to aggressively remove and replace senescent trees belonging to species that are not well adapted to a changing climate and produce relatively few benefits (e.g., high biogenic volatile organic compound emitters). Potential candidates include American sweetgum, velvet ash, tulip tree, London planetree and southern magnolia. By staging the removal of large senescent trees over a number of years, and removing trees that pose the greatest risk first, the destabilizing impacts of their loss on biomass and potential liability costs can be mitigated. Also, providing intensive care to extend the service life of the most desirable species (i.e., jacaranda and camphor) can promote biomass retention.

### 5. Conclusions

Urban forests can provide many important ecological functions and economic benefits, but continuous delivery of those services depends on the long-term health and resilience of the population. A multi-faceted approach to sustaining the desired level of urban forest services may include maintenance or increase of stocking levels by increasing plantings or encouraging growth of survivors, expansion of species and age diversity, and monitoring for changes in tree demography.

#### 5.1. Maintain or increase stocking level

Our longitudinal results indicate that while the removal rate was low, replacements did not keep up with removals. If this trend is mirrored by reduced plantings in previously vacant sites, stocking level will inevitably decline. Reduced plantings were largely a result of cuts to the municipal forestry budget. Prior to 2015, Claremont’s reforestation program was budgeted for 100 new trees per year (approximately 1 replacement for every 2 trees removed; J. Cawn, personal communication, Nov. 10, 2015). By 2015, the budget was raised to allow for closer to 300 new trees. Claremont stands out as having relatively high stocking (74%) but, as noted above, to maintain this level every removed tree should be replaced with a new tree. For major planting efforts, even-aged tree clusters (e.g. large plantings at the same time and of the same species) are not recommended (Sanders et al., 2013). A move towards uneven-aged structure will help to prevent pulses in canopy loss (best measured by tracking biomass) and maintain stocking level. Stabilization or increase of biomass stock can be achieved by appropriate maintenance of surviving trees (e.g., a species-appropriate pruning schedule to prevent premature loss, early pest detection and treatment when pests present a threat, aggressive removal and replacement of overmature trees belonging to less desirable species).

#### 5.2. Expand species and age diversity

Given that urban forests require large capital and long-term investments, thoughtful selection of tree species is important (Nowak and

Dwyer, 2007). The heterogeneous structure of urban tree populations with respect to age, size and species composition can offer resilience in the face of climate change and ebbs and flows in municipal funding. Diversity can help prevent disastrous consequences of pulses in mortality occurring in one segment of the population. For example, the impending threat from the Polyphagous Shot Hole Borer encourages planting of species not affected by the pest or to consider spatial patterns to buffer pest transfer (Eskalen et al., 2013). There can also be diversity in the stature of the mature species. In Claremont, the distribution by tree type is 55% large statured species, 20% medium and 25% small. Given the ecosystem services are typically greater from large statured trees, this might play to Claremont's advantage, depending on management objectives.

With respect to species diversity, Claremont fares better than many California cities (McPherson et al., 2016a). One current guideline recommends that no one species should account for more than 10% of the population's total importance value (IV), where IV incorporates the relative abundance, leaf area, and canopy cover of each tree species (McPherson and Kotow, 2013). All of Claremont's top tree species are below 9% IV, with American sweetgum the highest at 8.8% (Appendix A in Supplementary material). Even so, with the need for planting replacement trees and with impending losses of mature trees from senescence, there are opportunities to shift species composition to one that is more diverse and better adapted to future predicted scenarios. Experimenting with controlled trials of promising species is one way to ensure a steady supply of species options (McPherson et al., 2018).

### 5.3. Monitor for demographic changes

Our study demonstrates a technique for assessing urban street tree populations to estimate context and location-specific demographic rates. In the case of Claremont, we identify areas for improvement but also provide baseline information against which to compare changes. Results from future monitoring can be used to detect departure from baseline demographic rates.

Collecting longitudinal data requires funding, dedicated leadership and robust study design for statistical analysis (Lindenmayer and Likens, 2010). One way to minimize costs is to adapt existing investments in tree inventories so that they are better suited for longitudinal analyses. A critical step is to include geospatial coordinates, tagged tree identification numbers, or detailed site maps so that sample tree sites can be easily and accurately located (Roman, 2013). In subsequent years the same tree site should be assessed for tree survivorship and growth, as well as replacement (van Doorn et al., *In review*). Another inexpensive strategy is to gradually add newly planted sites to the sample for tracking through time.

The demographic analysis conducted in this study has helped identify the trajectory of the street tree population beyond just tree numbers and density, and can be used as input for projection models and corresponding ecosystem service calculations. Other municipal forestry programs could adopt this method to estimate removal, growth and replacement rates to gain a complete perspective of the state of their urban forest and to better understand where to focus management efforts.

### Acknowledgements

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### Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <https://doi.org/10.1016/j.ufug.2017.11.018>.

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